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WATER AND SEDIMENT TEMPERATURES AT MUSSEL BEDS IN THE UPPER MISSISSIPPI RIVER BASIN

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ABSTRACT

Native freshwater mussels are in global decline and urgently need protection and conservation. Declines in the abundance and diversity of North American mussels have been attributed to human activities that cause pollution, water-quality degradation, and habitat destruction. Recent studies suggest that effects of climate change may also endanger native mussel assemblages, as many mussel species are living close to their upper thermal tolerances. Adult and juvenile mussels spend a large fraction of their lives burrowed into sediments of rivers and lakes. Our objective was to measure surface water and sediment temperatures at known mussel beds in the Upper Mississippi (UMR) and St. Croix (SCR) rivers to estimate the potential for sediments to serve as thermal refugia. Across four mussel beds in the UMR and SCR, surface waters were generally warmer than sediments in summer, and were cooler than sediments in winter. This suggests that sediments may act as a thermal buffer for mussels in these large rivers. Although the magnitude of this effect was usually <3.0°C, sediments were up to 7.5°C cooler at one site in May, suggesting site-specific variation in the ability of sediments to act as thermal buffers. Sediment temperatures in the UMR exceeded those shown to cause mortality in laboratory studies. These data suggest that elevated water temperatures resulting from global warming, thermal discharges, water extraction, and/or droughts have the potential to adversely affect native mussel assemblages.

KEY WORDS Native freshwater mussels, Water temperature, Mississippi River, Unionids, Climate change

INTRODUCTION

Anthropogenic warming is changing thermal regimes in freshwater systems. The effects of climate change have been seen in nearly every ecosystem; however, aquatic systems may be especially sensitive to thermal stress because of human alterations such as dams and diversions, deforestation, urbanization, and channelization (Hester & Doyle, 2011). In aquatic systems, climate change can alter thermal regimes, reduce ice cover, change stream flows, increase water development, and increase salinity (Rahel & Olden, 2008). It is well established that elevated temperatures can adversely affect aquatic organisms. For example, elevated water temperatures have been associated with increased energy requirements of young-of-the-year fishes (McDonald et al., 1996), reduction in available habitat for stream biota (Eaton & Scheller, 1996), increased probability of outbreaks of toxic algal blooms (Gilbert, 1996), and more rapid life cycle completion in stream invertebrates (Wilhelm & Schindler, 2000).

Native freshwater mussels are long-lived, benthic filter-feeding organisms that provide important ecological services to aquatic systems (Vaughn & Hakenkamp, 2001; Spooner & Vaughn, 2008). Mussels are frequently found in dense, species-rich assemblages called mussel beds. However, many river systems have lost substantial numbers of native mussel species in the past century. For example, about 20 mussel species have been functionally lost from the Upper Mississippi River basin, and many others are state or federally listed (Newton et al., 2011). Losses in species richness and biomass appear to result from varied anthropogenic effects including impoundments, water management alterations, invasive species, changing land use, pollution and most recently, climate change (Hastie et al., 2003; Strayer et al., 2004; Galbraith et al., 2010).

The mechanisms by which elevated water temperatures may influence mussel assemblages are poorly known, largely because thermal tolerance has been studied for few species. To our knowledge, quantitative data on lethal temperatures is limited to <15 species (~5% of the 300 known species in North America). Most studies on thermal tolerance in mussels result from acute laboratory studies with early life stages. These studies generate an LT50 which is the median lethal temperature that causes mortality in 50% of the individuals over a specified time interval. For example, 4-d LT50s across 11 species of juveniles ranged from 32.5 to 38.8°C (Pandolfo et al., 2010; Archambault et al., 2012). In chronic tests, Ganser et al. (in press) observed 28-d LT50s that ranged from 25.3 to 30.3°C among three species of juveniles.

Considerably less is known about the thermal tolerance of adult mussels. The maximum temperature at which five species of mussels were observed in the River Rhine ranged from 24 to 28°C, even though water temperatures can reach 32°C in this system (Verbrugge et al., 2012). Estimated critical thermal maxima (the temperature at the onset of behavioral incapacitation) in three species of mussels (*Alasmidonta varicosa*, *Elliptio complanata*, and *Strophitus undulatus*) ranged from 39.1 to 42.7°C (Galbraith et al., 2012). Bartsch et al. (2000) suggest that adults of three species (*Elliptio dilatata*, *Quadrula pustulosa*, *Lampsilis cardium*) were remarkably resistant to thermal shock.

Although vertical movement into sediments has been described as an important behavior in mussels (Haag, 2012), we know little about this behavior — especially in rivers. Adult mussels burrow as deeply as 25 cm, but usually burrow <10 cm (Balfour & Smock, 1995; Schwalb & Pusch, 2007; T.J. Newton, unpublished data). We know considerably less about burrowing activities in juveniles. In captivity, juveniles typically burrowed <1 cm (Yeager et al., 1994). Burrowing behavior often varies with biological (e.g., reproduction, Amyot & Downing, 1998; Eads & Levine, 2013) and environmental parameters (e.g., season, flow, substrate, Di Maio & Corkum, 1997). Many species of adults can be found near the sediment surface in spring and summer but may burrow more deeply in fall and winter (Amyot & Downing, 1997; Schwalb & Pusch, 2007). While adult mussels exhibit vertical migration patterns in the sediment with periods at, above, or below the sediment surface, juveniles appear to remain burrowed in sediments for the first few years of life (Cope et al., 2008).

The ability to accurately assess the thermal tolerances of multiple life stages of mussels in the wild is limited by the inadequate understanding of the background thermal regimes in river sediments — the environment in which mussels reside for most of their lives. Our objective was to measure surface water and sediment temperatures at known mussel beds in the Upper Mississippi (UMR) and St. Croix rivers (SCR) to estimate the potential for sediments to serve as refugia during times of thermal stress.

METHODS

We selected four mussel beds in the UMR and four beds in the SCR that had high mussel density, high species richness, and contained a range of age classes including young individuals of several species (Fig. 1). The beds at sites 1-6 were located in the border of the main navigation channel, while the beds at sites 7 and 8 were in large side channels. The mussel beds ranged from ~22,000-222,000 m^2 in size. The substrate was predominately medium to coarse sands in all beds. These sites are representative of areas where dense and diverse mussel assemblages typically occur in these rivers.

We placed submersible temperature data loggers (iBCod, Alpha Mach, Inc., Mont St-Hilaire, Quebec, Canada) at 5-7 locations in each mussel bed (Table 1). The locations were chosen to span the area encompassed by each mussel bed. Temperature loggers were mounted on Trex® composite stakes in a manner that allowed them to be deployed in three vertical strata at a single point: 5 and 15 cm below the sediment-water interface and in the water column, 10 cm above the sediment-water interface (hereafter referred to as surface water stratum, Fig. 2). Due to cost, water column loggers were placed on only two of the samplers (selected at random) at each site. Temperature loggers were initially deployed in the summer of 2010; retrieved, downloaded and re-deployed in the fall of 2010 and spring of 2011; and retrieved and downloaded in the fall of 2011. Due to limited memory, temperature loggers were programmed to record temperatures hourly in the summer and fall and every three hours in the winter and spring. Because retrieval rates of temperature loggers were low (see below), detailed statistical analyses were not conducted. Rather, we examined the data for patterns in water and sediment temperatures over time and among depth strata. We estimated the deviation between surface water and sediment temperatures as the difference in temperature between surface water and the 5 or 15 cm sediment depth. If the deviation was >0, sediment temperatures were cooler than surface water temperatures.

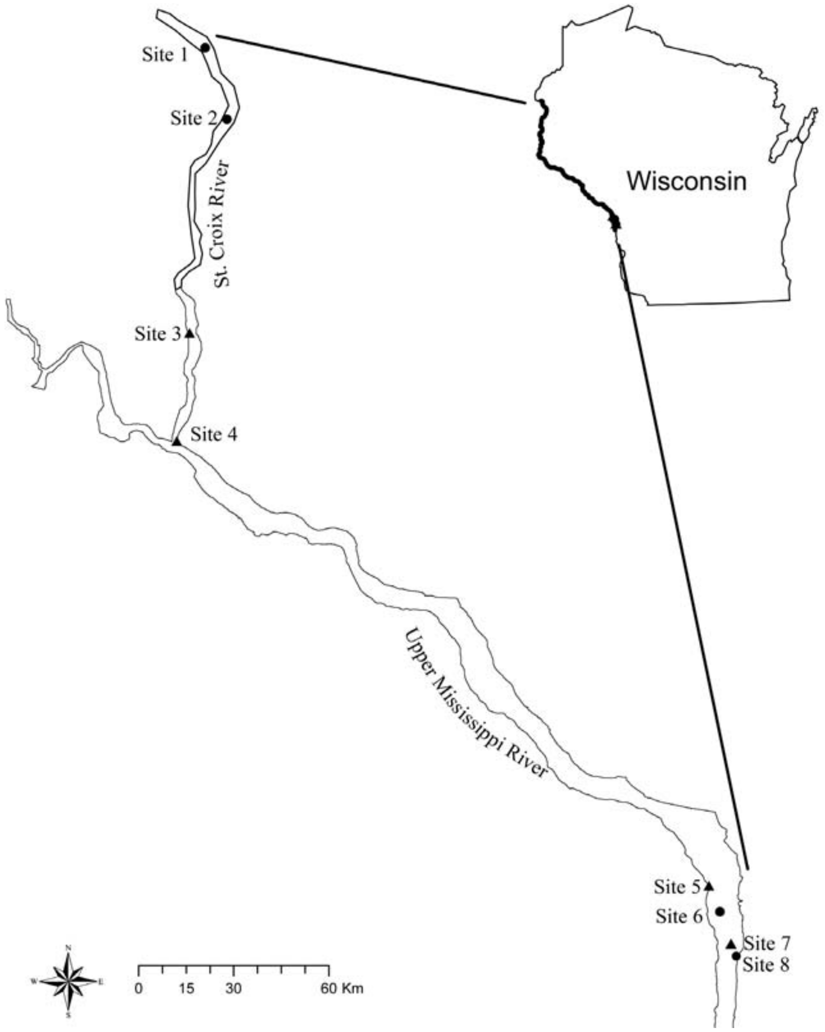

FIGURE 1

Map of locations where temperature recording data loggers were deployed in the St. Croix and Upper Mississippi rivers. Sites denoted by triangles had at least 10 months of surface water and sediment temperature data.

TABLE 1

Deployment of temperature data loggers (i.e., iBCod's) at known mussel beds in the St. Croix (sites 1-4) and Upper Mississippi (sites 5-8) rivers. Temperature loggers were deployed on stakes and each stake contained a data logger at 5 and 15 cm below the sediment-water interface. Each site also had two randomly placed data loggers that were 10 cm above the sediment-water interface.

Site	No. stakes deployed	Total no. iBCod's deployed	Date initially deployed[a]	Date of final retrieval	No. of iBCod's with <10 months of data	No. of iBCod's with ≥10 months of data
1	7	16	7-23-2010	7-20-2011	10	0
2	7	16	7-23-2010	7-20-2011	9	0
3	7	16	8-19-2010	10-4-2011	9	3
4	7	16	8-19-2010	10-5-2011	12	1
5	7	16	7-29-2010	10-20-2011	5	4
6	5	12	7-29-2010	10-20-2011	2	0
7	5	12	8-11-2010	10-19-2011	3	2
8	7	16	7-29-2010	10-19-2011	4	1

[a]sampling interval during summer and fall deployment was 1 hour; sampling interval during winter and spring deployment was 3 hours

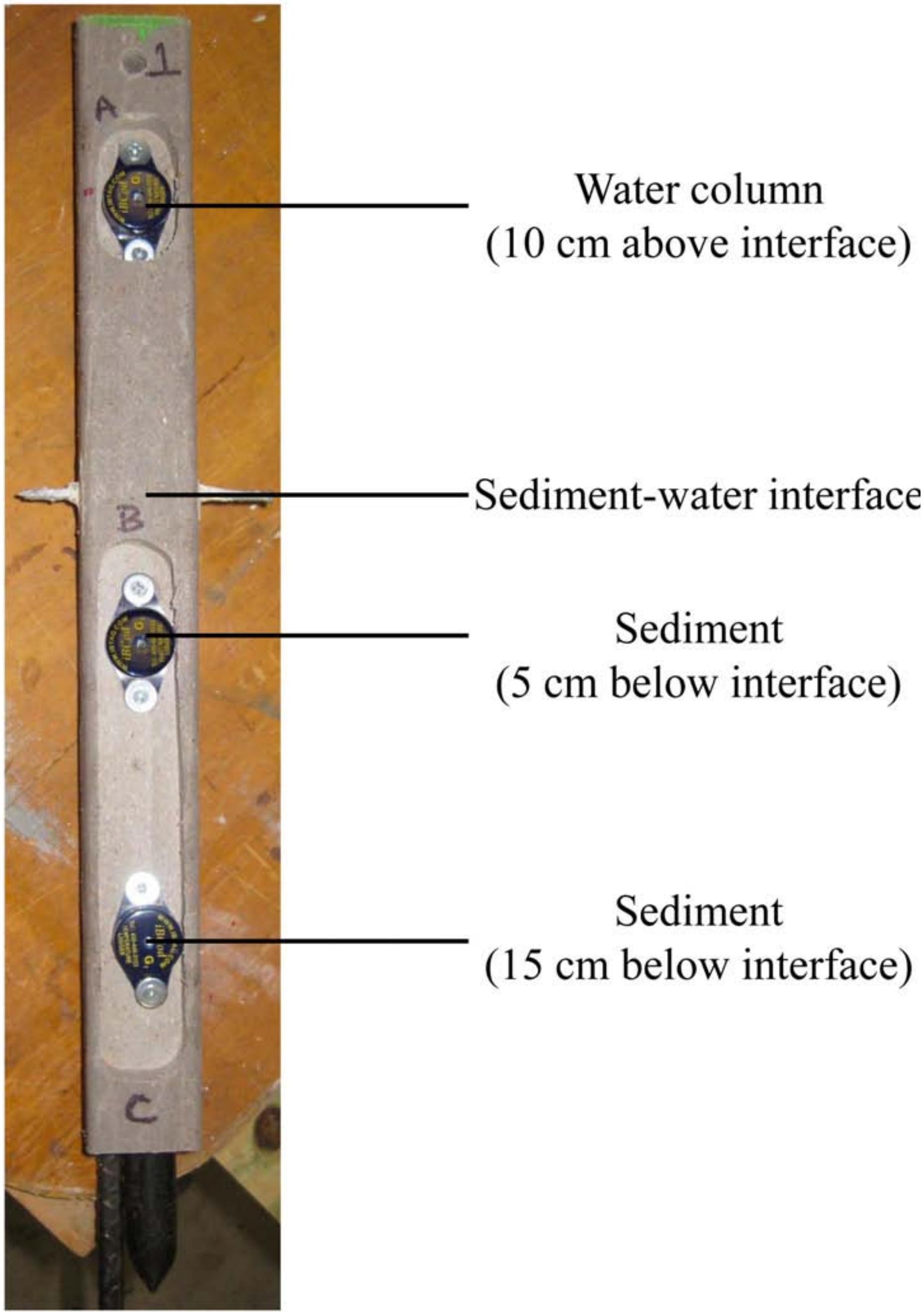

FIGURE 2

Schematic of stakes used to deploy temperature recording data loggers in the St. Croix and Upper Mississippi rivers.

RESULTS

We obtained data from few temperature loggers because retrieval rates were low due to high water and nearly half of the retrieved loggers experienced electrical malfunction (Table 1). However, we have at least 10 months of data for at least two of the depth strata from one stake at two mussel beds on the SCR and two mussel beds on the UMR (Table 2). Across these sites, surface waters were generally warmer than sediments from spring through fall and cooler than sediments in winter (Figs. 3 and 4). Temporal patterns were similar across sites, although sites in the UMR were ~1-2°C warmer in summer than sites in the SCR. From fall 2010 through spring 2011, mean temperatures ranged from 0 to 26°C in the surface water and 5 cm sediment stratum and from 0 to 25°C in the 15 cm sediment stratum (Table 3).

TABLE 2

Dates over which surface water and sediment temperature data were available at sites with known mussel beds in the St. Croix (SCR) and Upper Mississippi (UMR) rivers. Temperature loggers were deployed 10 cm above the sediment-water interface (surface) and at 5 and 15 cm below the sediment-water interface.

Site (River)	Depth strata	Dates	Approximate time interval
3 (SCR)	surface, 5 and 15 cm	8-20-2010 to 10-28-2010	2 mon
	surface and 5 cm	8-20-2010 to 6-6-2011	10 mon
4 (SCR)	surface, 5 and 15 cm	8-20-2010 to 6-12-2011	10 mon
	5 cm	8-20-2010 to 8-11-2011	12 mon
5 (UMR)	surface, 5 and 15 cm	7-30-2010 to 10-20-2011	15 mon
7 (UMR)	surface, 5 and 15 cm	8-12-2010 to 10-21-2010	2 mon
	5 and 15 cm	8-12-2010 to 7-5-2011	11 mon

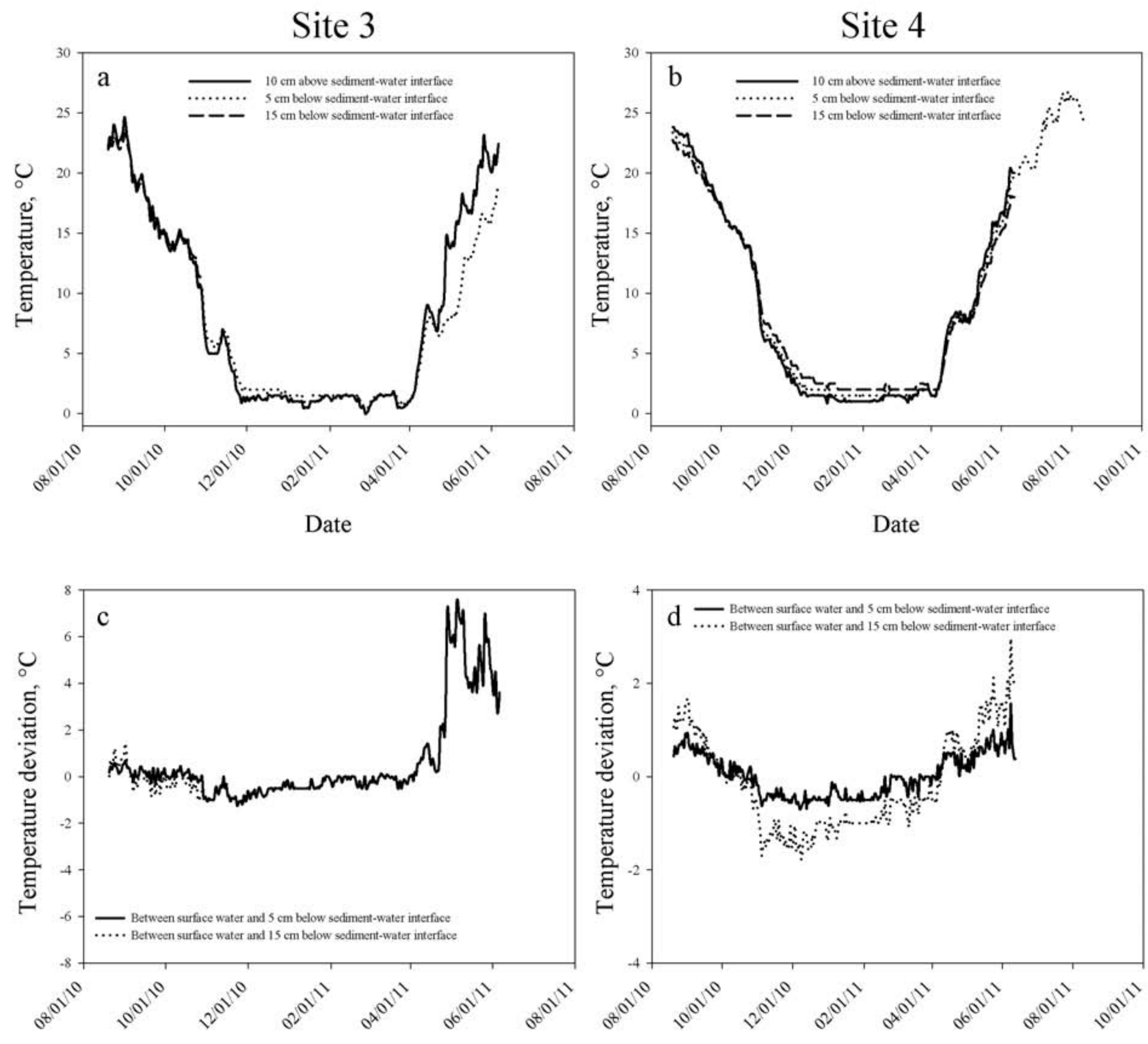

FIGURE 3

Temperature patterns in three strata (10 cm above the sediment-water interface and 5 and 15 cm below the sediment-water interface) and temperature deviations between surface and sediment strata from one stake at two sites with known mussel beds in the St. Croix River. A positive temperature deviation indicates that surface waters were warmer than sediments.

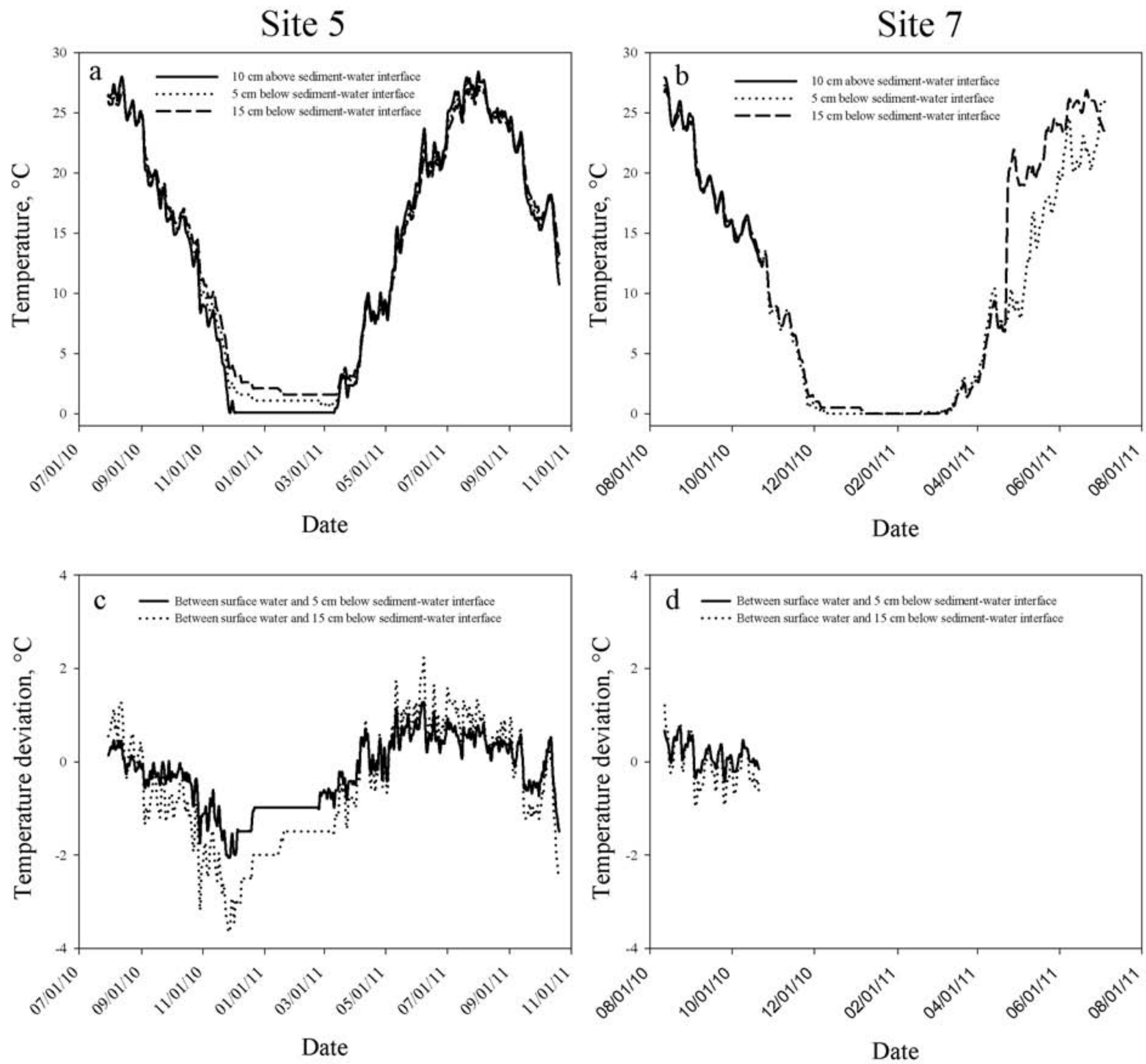

FIGURE 4

Temperature patterns in three strata (10 cm above the sediment-water interface and 5 and 15 cm below the sediment-water interface) and temperature deviations between surface and sediment strata from one stake at two sites with known mussel beds in the Upper Mississippi River. A positive temperature deviation indicates that surface waters were warmer than sediments.

Temperature deviations were variable among sites, ranging from 2.1°C warmer to 7.5°C cooler in the 5 cm stratum and from 3.6°C warmer to 2.9°C cooler in the 15 cm stratum relative to surface water (Figs. 3 and 4). On average, temperatures in the 5 cm stratum were 0.2 to 1.0°C warmer during fall, winter and spring and 0.5-4.0°C cooler during summer compared to surface water (Figs. 3 and 4). However, we observed a substantial deviation between surface water and the 5 cm sediment stratum (+7.5°C) at site 3 in the spring 2011 (Fig. 3c), presumably due to rapidly rising water temperatures in the spring (Fig. 3a). On average, sediment temperatures were 0.8 to 1.8°C warmer during fall, winter and spring and 0.6-0.7°C cooler during summer in the 15 cm stratum relative to surface water (Figs. 3 and 4).

We hypothesized that surface water temperatures might be more variable than sediment temperatures, however, we found little evidence for this in the present study. For example, over the time period of July 2010 to July 2011 at site 5, the mean coefficient of variation (CV) of temperatures in surface water was 69%. Similarly, the CV was 65% and 61% at 5 and 15 cm below the sediment-water interface, respectively.

TABLE 3

Descriptive statistics of surface water and sediment temperature (°C) at sites with known mussel beds in the St. Croix (SCR) and Upper Mississippi (UMR) rivers during August 20, 2010 to June 5, 2011. Temperature loggers were deployed 10 cm above the sediment-water interface (surface) and at 5 and 15 cm below the sediment-water interface. Deviation is the difference in temperature between surface water and the 5 or 15 cm sediment depth. If a given depth stratum is not listed, there were not data over the entire time interval.

Depth stratum	Mean	SD	Range
		Site 3 (SCR)	
surface	8.2	7.9	0 to 24.6
5 cm	7.6	7.0	0.5 to 23.9
5 cm deviation	0.6	2.0	-1.3 to 7.5
		Site 4 (SCR)	
surface	8.0	7.4	0.9 to 23.8
5 cm	8.0	7.1	1.2 to 23.4
15 cm	8.2	6.7	2.0 to 22.8
5 cm deviation	-0.1	0.4	-0.7 to 1.0
15 cm deviation	-0.2	1.0	-1.8 to -2.1
		Site 5 (UMR)	
surface	7.9	8.0	0.1 to 26.0
5 cm	8.4	7.5	0.7 to 25.9
15 cm	8.9	7.3	1.6 to 25.4
5 cm deviation	-0.5	0.7	-2.1 to 1.1
15 cm deviation	-1.1	1.2	-3.6 to 1.8
		Site 7 (UMR)	
5 cm	7.8	7.9	0 to 25.2
15 cm	9.0	9.0	0 to 25.1

DISCUSSION

Although the data set is limited, our data suggest that river sediments may act as a thermal buffer for native mussels during winter and summer in mussel beds in the UMR basin. In summer, temperatures were 0.5 to 40°C cooler in the 5 cm sediment stratum which would provide mussels a refuge from warm summer temperatures, which may be important during this time of active movement and reproduction in many species. In winter, warmer temperatures in sediments (range, 0.2-1.8°C) may allow mussels to live at temperatures closer to groundwater and provide a refuge from cold winter temperatures, especially in shallow waters. The ability of mussels to move vertically in response to temperature has been observed in other studies. In mesocosms, *Actinonaias ligamentina* burrowed into sediments during periods of high water temperatures, presumably to seek out cooler interstitial waters (Allen & Vaughn, 2009). Amyot & Downing (1997) reported that vertical migration of *Elliptio complanata* in a Canadian lake was significantly correlated with water temperature. However, the ability of sediments to act as thermal buffers may be site-specific and more research on those variables (e.g., particle size, ground water influence, water content) that influence vertical thermal profiles is needed.

Although the magnitude of the differences between surface water and sediment temperatures may not seem large, laboratory studies have shown that the average difference between temperatures that killed 5% (LT05) and 50% (LT50) of juveniles was only 4-5°C (Pandolfo et al., 2010; Ganser et al., in press). Given that most of our sites were in channel border areas characterized by coarse sand and some hyporheic flow, we might not expect to see much variation in surface water temperature among depth strata. The fact that we observed up to a 7°C differential between surface water and sediment temperature in a river as large as the UMR, with high thermal inertia, suggests that similar differences in smaller rivers may be considerable. Thus, small changes in sediment temperatures (relative to surface waters) may provide mussels an opportunity to alleviate thermal stress.

The temperatures observed in sediments in mussel beds in the UMR basin can exceed those shown to cause mortality in the laboratory. For example, chronic laboratory exposures of three species of juveniles resulted in 28-d LT50s that were 25.3°C in *Lampsilis siliquoidea*, 27.2°C in *Lampsilis abrupta*, and 30.3°C in *Megalonaias nervosa* (Ganser et al., in press). A sediment temperature of 25°C was exceeded 37-64% of the time and 27°C was exceeded 10-26% of the time during summer at one site in the UMR (Fig. 5). During the summer of 2006, the UMR experienced exceptionally low flows and high water temperatures which resulted in 31 days with sediment temperatures >29°C, 16 days with temperatures >30°C and 8 days with temperatures >31°C downstream of a thermal discharge (Dunn, 2009). Thus, temperatures that cause chronic mortality to juveniles in

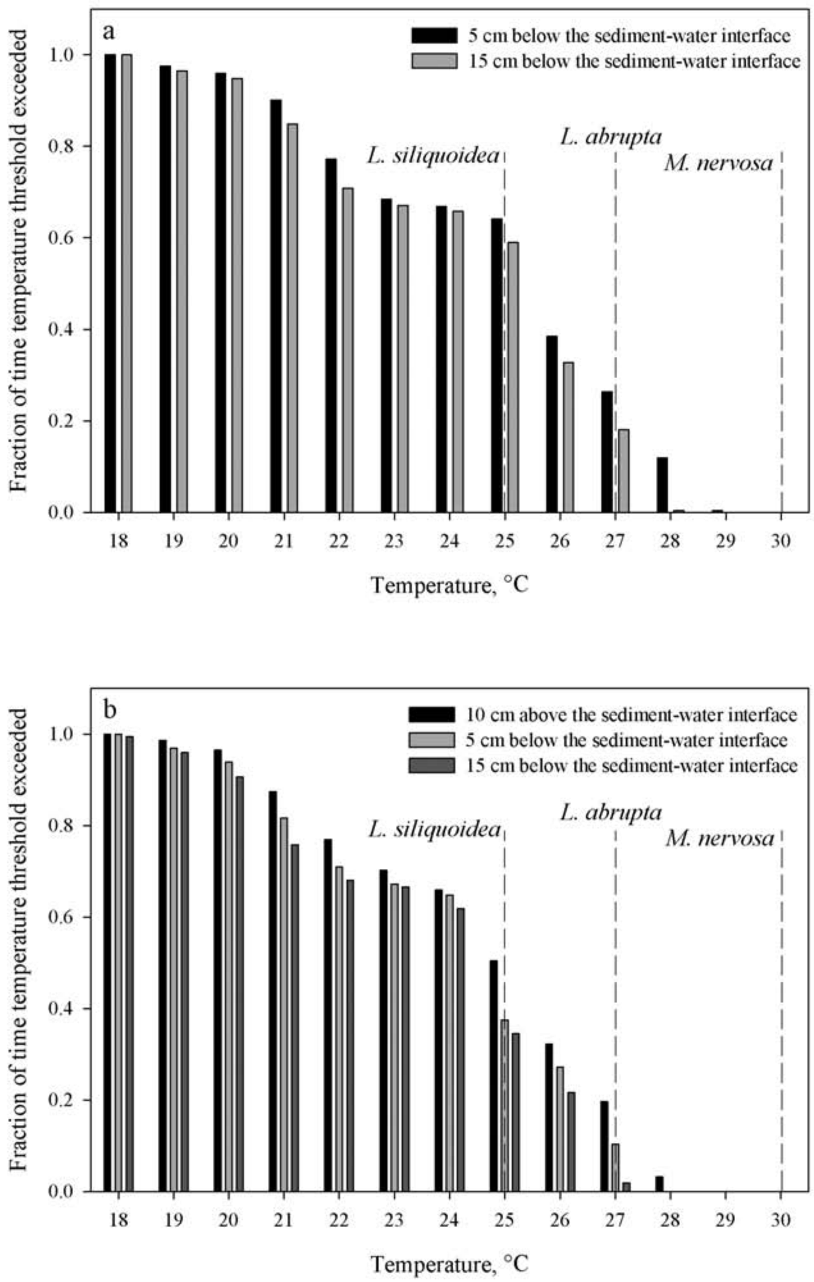

FIGURE 5

The fraction of time that a given river temperature was exceeded at Site 5 in the Upper Mississippi River. Dashed vertical lines represent the 28-day lethal temperature that resulted in 50% mortality in three mussel species (*Lampsilis siliquoidea*, *Lampsilis abrupta*, and *Megalonaias nervosa*) in laboratory studies (Ganser et al., in press). Temperatures were recorded every 3 hours during June, July and August 2011 at an upstream (a) and downstream (b) stake in this mussel bed.

laboratory studies can be exceeded for lengthy periods of time during summer in the UMR. Although most thermal tolerance data on juveniles are derived from water-only tests which may not accurately represent their benthic nature, a recent study showed that the addition of sediment allowing juveniles to burrow, did not offer any thermal protection in acute tests (Archambault et al., 2012).

We hypothesized that temperatures in surface waters might be more variable than in sediments (due to diel warming and cooling), although we have no evidence to support this. The lack of such an effect could result from our limited sample size or the fact that our surface water samples were taken from near the sediment-water interface. In a system as deep and well mixed as the UMR, our surface water temperatures may be more representative of sediment temperatures. Regardless, we observed considerable heterogeneity in surface water temperatures across all depth strata. This variation suggests that mussels may not need to move far to reach different temperatures. This might be especially important in juveniles — a life stage that spends much of their first few years buried in river sediments (Cope et al., 2008) and for which we know little about movement patterns. Such heterogeneity may create thermal refugia and mitigate some of the potential negative effects of temperature on mussels (Verbrugge et al., 2012).

Despite its limitations, this study increases our understanding of the potential effects of elevated river temperatures on native mussel assemblages and on the potential for sediments to provide a thermal buffer in rivers. Data on the thermal biology of native mussels are needed to help conserve and restore native mussel populations and to forecast species responses to climate change over the next few decades. Management actions such as the creation of thermal buffers in riparian zones and maintenance of sufficient flows during critical life history periods might reduce the effects of elevated temperatures on native mussel assemblages.

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RECENT COLLECTION OF A FALSE SPIKE (*QUADRULA MITCHELLI*) IN THE SAN SABA RIVER, TEXAS, WITH COMMENTS ON HABITAT USE

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ABSTRACT

Similar to other rare and endemic freshwater mussel species in Texas, the distribution and life history of the False Spike, *Quadrula mitchelli*, is poorly understood. Few recent locality records suggest that *Q. mitchelli* has been extirpated from much of its range and is declining in numbers at an alarming rate, which has led to it being petitioned for listing under the Endangered Species Act. We present our findings of the discovery of one live individual collected on the San Saba River, TX and provide information regarding the species' habitat use. The discovery represents the second known population in Texas and the only record of a live individual from the San Saba River. Knowledge of habitat use may help identify populations in other streams and allow managers to develop recovery plans for *Q. mitchelli*. However, given the rarity of this species, *Q. mitchelli* potentially faces extinction unless prompt conservation action is taken by state and federal agencies.

KEY WORDS Freshwater mussels, Unionids, Texas, False Spike, Rare Species

INTRODUCTION

In Texas, the current status and life history of rare freshwater mussel (Bivalvia: Unionidae) species are poorly understood. False Spike, *Quadrula mitchelli* (Simpson, 1895), is a rare species of mussel endemic to Central Texas and the Rio Grande drainage. Strecker (1931) described the species as being common wherever it was found. Over the last 30 years, considerable effort has been spent trying to locate live individuals of this species in stream segments where it historically occurred, but to date contemporary accounts are largely based on weathered shell material (Howells, 2010). The only exceptions are the discovery of several live individuals of *Q. mitchelli* in the Guadalupe River and a fresh dead individual in the San Saba River (Randklev et al., 2012; Randklev et al., in press). These accounts indicate that *Q. mitchelli* is still extant, but the absence of this species in other parts of its range suggest *Q. mitchelli* is no longer common, an observation reported as early as the 1970s (Stansbery, 1971). The decline of this species in stream segments where it once occurred has been attributed to anthropogenic impacts to streams and rivers coupled with record droughts and floods in the late 1970s and early 1980s in Texas (Howells, 2003). As a result, *Q. mitchelli* was designated as state threatened in 2009 (Texas Register 35 2010) and is currently being reviewed for listing under the Endangered Species Act (ESA; Federal Register 76 2011).

Historically, the range of *Q. mitchelli* included the Rio Grande, San Antonio, Guadalupe, Colorado, and Brazos river basins (Howells et al., 1996; Howells, 2010). The species is likely extirpated from the Rio Grande drainage (Howells, 2003). In central Texas, a single subfossil valve was collected from Salado Creek (Howells, 2002), representing the only record of the species in the San Antonio River drainage. Within the Guadalupe River drainage, Strecker (1931) and Wurtz (1950) collected live individuals from the Guadalupe River in the early decades of the twentieth century. Valves of a recently dead individual (shells in good condition, but soft tissue absent; Howells, 2003) were collected in the San Marcos River, a major tributary of the Guadalupe River, in 2000 (Howells, 2001). *Quadrula mitchelli* has been historically collected from Pecan Bayou (R.G. Howells database), Johnson Fork Creek (R.G. Howells database), Pedernales (Howells, 1994), San Saba (Strecker, 1931; Howells, 1995), and Llano (Strecker 1931; Howells 1996) rivers within the Colorado River drainage and from the Brazos (R.G. Howells database), Lampasas (R.G. Howells database), and Leon rivers (Strecker, 1931; R.G. Howells database) within the Brazos River drainage. Until recently, however, the only evidence to suggest the species still exists in Texas was the discovery of a valve of a fresh-dead individual (soft tissue present) from the San Saba River (Randklev et al., in press) and seven live individuals collected in the Guadalupe River in 2011 (Randklev et al., 2012).

METHODS

In July 2012, we conducted multiple-pass-depletion surveys for state-threatened mussel species in the lower San Saba River as part of a larger, ongoing study in the river. While conducting timed searches, we collected one live *Q. mitchelli* (Fig. 1) at a site located 11.3 km east of San Saba, San Saba Co., Texas, approximately 200 m upstream from a fresh-dead specimen reported by Randklev et al. (in press). Gonadal fluid was extracted to determine sex and reproductive viability (Saha & Layzer, 2008).

To improve our understanding of *Q. mitchelli's* habitat, we recorded physical measurements of habitat at the site. Six equidistant cross-section transects along the length of the site (76.5 m) were used to determine site-specific habitat characteristics. We measured depth (m) and velocity (ms^{-1}) at 0.5 m increments along each transect to determine mean discharge (m^3s^{-1}). Pebble counts (Wolman, 1954) were conducted along each transect to determine median substrate particle size (Gordon et al., 2004). Additionally, three 0.25-m^2 quadrats, placed on and directly adjacent to where the *Q. mitchelli* individual was collected, were used to determine microhabitat characteristics. We measured depth, velocity, and substrate characteristics (dominant, subdominant, and percentage fine sediment) for each of the quadrats mentioned above. We measured shear stress with FST hemispheres (Statzner & Müller, 1989) and visually determined percentage of benthic algae within each quadrat. Canopy cover (%) was visually estimated by three observers standing over the middle quadrat.

FIGURE 1
Live individual of *Quadrula mitchelli* collected from the San Saba River, San Saba Co., Texas.

RESULTS

The individual collected was consistent with taxonomic descriptions provided by Howells (2010) and measured 68.4 mm in shell length, representing a large adult. The presence of eggs in gonadal fluid extracted from the visceral mass clearly indicated that the individual was a viable female capable of reproducing, but the individual appeared not to be gravid at the time of sampling. Timed searches were conducted for a total of 70.6 person-hours (p-h) at the site, with an overall catch-per-unit-effort of 13.1 mussels collected per p-h of search effort (Table 1). Eight species were collected throughout the site during our survey, including four species listed as state threatened (Table 1; Texas Register 35 2010). Of these, *Q. houstonensis*, *Q. petrina*, and *Truncilla macrodon* are listed as candidates for protection under the Endangered Species Act (Federal Register 76 2011).

TABLE 1

Freshwater mussel species collected from one site on the San Saba River, San Saba Co., Texas where one live individual of *Quadrula mitchelli* was observed. Total individuals collected and catch-per-unit-effort (CPUE) are provided.

Species	Common Name	Live Individuals Collected
Amblema plicata (Say 1817)	Threeridge	8
Cyrtonaias tampicoensis (I. Lea 1838)	Tampico Pearlymussel	1
Leptodea fragilis (Rafinesque 1820)	Fragile Papershell	21
**Quadrula houstonensis* (I. Lea 1859)	Smooth Pimpleback	390
**Quadrula mitchelli* (Simpson 1859)	False Spike	1
**Quadrula petrina* (Gould 1855)	Texas Pimpleback	247
Quadrula verrucosa (Rafinesque 1820)	Pistolgrip	251
**Truncilla macrodon* (I. Lea 1859)	Texas Fawnsfoot	3
Total individuals collected		922
Total person-hours of effort		70.6
CPUE (mussels/p-h)		13.1

[a]State-threatened species and species being reviewed for potential listing under the Endangered Species Act (ESA; Federal Register 76 2011).

The site inhabited by the *Q. mitchelli* individual was characterized by steep banks with extensive riparian vegetation and adjacent land uses comprised of pecan orchards and rangeland. The channel was relatively wide and shallow with gravel and cobble substrates and moderate to high water velocities (Table 2). The macrohabitat of the site consisted of a run-riffle-pool sequence. We collected the individual in a run, immediately upstream from where flow transitioned into a riffle, and *Q. mitchelli* was observed burrowed in very coarse gravel. Benthic algae were relatively abundant on gravel and cobbles and on the shells of live mussels.

DISCUSSION

Limited information regarding habitat preferences for *Q. mitchelli* exists. Wurtz (1950) and Randklev et al. (2012) collected live specimens in the Guadalupe River at sites with relatively shallow depths (< 0.7 m) with gravel and cobble substrates, similar to habitat observed in the present study. However, Wurtz (1950) noted water lilies at a site where *Q. mitchelli* was present but no other study reported macrophytic vegetation where the species was collected.

While only one individual was collected during our survey, the physical habitat measurements we recorded might be of use in locating similar habitat for *Q. mitchelli* in streams within the Colorado River basin. Also, if this species becomes listed under the ESA, our observations of habitat might assist resource managers with mapping of critical habitat. Finally, there is little information available on the effort needed to collect this species by using the timed-search method. Therefore, the amount of effort we invested to locate a live individual of *Q. mitchelli* can be used to guide surveys targeting this species, especially in streams where it is suspected to occur in low densities.

The status of *Quadrula mitchelli* in Texas, based on historical and contemporary surveys, is tenuous. The

TABLE 2

Site-specific habitat and microhabitat estimates collected from San Saba River, San Saba Co., Texas where one live individual of *Quadrula mitchelli* was collected. Microhabitat refers to habitat measured from quadrats on and adjacent to the area where *Q. mitchelli* was collected.

Habitat parameters	Estimate
Site-specific habitat[a]	
Mean bankfull width	23.52 m ± 1.42 (SD)
Mean bankfull depth	0.73 m ± 0.04 (SD)
Mean wetted width	22.63 m ± 1.27 (SD)
Mean depth	0.20 m ± 0.04 (SD)
Discharge range	1.42 – 1.81 m^3s^{-1}
Median substrate particle size	36 – 67 mm
Microhabitat[b]	
Mean current velocity	0.51 ± 0.02 m s^{-1} (SD)
FST-hemisphere density	1.129 – 1.274 g cm^{-3}
Mean dominant substrate	119.17 mm ± 24.18 (SD)
Mean subdominant substrate	58.60 mm ± 3.33 (SD)
Mean fine substrate	11.67% ± 6.24 (SD)
Mean benthic algae	46.67% ± 4.71 (SD)
Mean canopy cover	69% ± 5.66 (SD)

[a]n = 6 for all estimates. bn = 3 for all estimates.

species appears to have been extirpated from much of its range and until our discovery in the San Saba River has only been recently collected alive from the Guadalupe River (Randklev et al., 2012). Given the amount of time expended to collect this individual (70.6 p-h) and the fact that we observed this species at only one site, despite surveying other locations with similar effort, does not bode well for *Q. mitchelli* in the San Saba River. Thus, it appears that *Q. mitchelli* is on the brink of local extinction in this river, which is problematic because it is only known to persist at one other locality. Currently, there are no substantive plans to mitigate the decline of this species which indicates to us that the likelihood of recovering *Q. mitchelli* is low unless prompt conservation action is taken by Texas Parks and Wildlife Department and U.S. Fish & Wildlife Service.

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STATUS OF FRESHWATER MUSSELS IN THE MIDDLE FORK HOLSTON RIVER, VIRGINIA

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ABSTRACT

Six sites in the Middle Fork Holston River (MFHR), Virginia, were surveyed in 2010 and 2011 using catch-per-unit-effort (CPUE, no./h) and 0.25 m^2 quadrats to assess changes in the mussel fauna since a previous survey in 1998. Since 1998, species richness declined from 15 to 11, compared to a historical richness of at least 20 species. To date, extirpated species are dominated by short-lived species, but all remaining species are declining rapidly. Mussel abundance, both as density (number/m^2) and CPUE, declined by ≥50% since 1998 at most sites, and several species collected during this study were represented by only a few individuals. There was no evidence of recent reproduction at the survey sites. Although the federally endangered *Epioblasma florentina aureola* appears to be extirpated, two species proposed for federal listing, *Pleuronaia dolabelloides* and *Ptychobranchus subtentum*, persist in the river. The MFHR appears to be another example of an enigmatic mussel decline characterized by curtailed recruitment and subsequent erosion of the fauna over time, despite a lack of obvious impacts to the stream. Twenty-six reaches in the MFHR watershed are listed as impaired, primarily by sediment and *E. coli*, suggesting that nutrient enrichment coupled with increases in streambed embeddedness could produce elevated substrate ammonia concentrations, which are toxic to juvenile mussels. In addition, limited sediment quality data indicate that metals, PCBs, and DDE are present in the stream and also may inhibit recruitment or have sublethal effects on adult mussels. The MFHR is an important refuge for the diverse Tennessee River basin mussel fauna, and identification and remediation of specific factors responsible for mussel declines are urgently needed.

KEY WORDS Freshwater mussels, Unionidae, Middle Fork Holston River

INTRODUCTION

The Middle Fork Holston River (MFHR) is a tributary of the Tennessee River system, and it historically supported a freshwater mussel fauna of at least 20 species (Henley et al., 1999). The MFHR potentially is an important conservation refuge for the unique mussel fauna of this region, and it previously supported one of only two remaining populations of *Epioblasma florentina aureola*; however, the fauna of the river has declined substantially in recent decades. By 1998, only 15 species were collected from the river, but abundances were low for most species and evidence of recent recruitment was absent at nearly all sites (Henley et al., 1999). These observations suggest a steady decline in mussel diversity and abundance throughout the river. Mussel abundance was exceptionally low downstream of the towns of Atkins, Marion, and Chilhowie, indicating possible effects of point source discharges. We resurveyed the MFHR in 2010 and 2011 to assess the current status of the mussel fauna with particular emphasis on documenting changes over the 12 years since the survey of Henley et al. (1999).

METHODS AND MATERIALS

The MFHR flows southwest through Wythe, Smyth, and Washington counties, southwestern Virginia, to its confluence with the South Fork Holston River near Abingdon (Fig. 1). The watershed lies within the Ridge and Valley physiographic province and is underlain primarily by limestone bedrock (Henley et al., 1999). Average discharge for the period between 1932 and 2010 at the USGS gaging station near Meadowview, Virginia, was 6.9 (±1.9 SD) cms, with a mean monthly summer flow of 3.8 (±2.6) cms and mean annual peak flow of 131.2 (±72.8) cms (USGS, 2012). The watershed covers approximately 625 km^2, and current land uses are roughly 49% forest, 41% pasture, 9% residential, and 1% cropland (USEPA, 2010). About 8 km of the stream are impounded by Edmonson Dam, located at MFHRKM 22.5, and approximately 2.4 km of the lower river are influenced by South Holston Lake. There is a small (approximately one m high), damaged dam at MFHRKM 32.7 in Chilhowie, Smyth County, that probably does not inhibit fish passage. Also, there is an approximately 2.4 m high milldam at MFHRKM 31.6 at DeBusk Mill, Washington County. This milldam does inhibit fish passage.

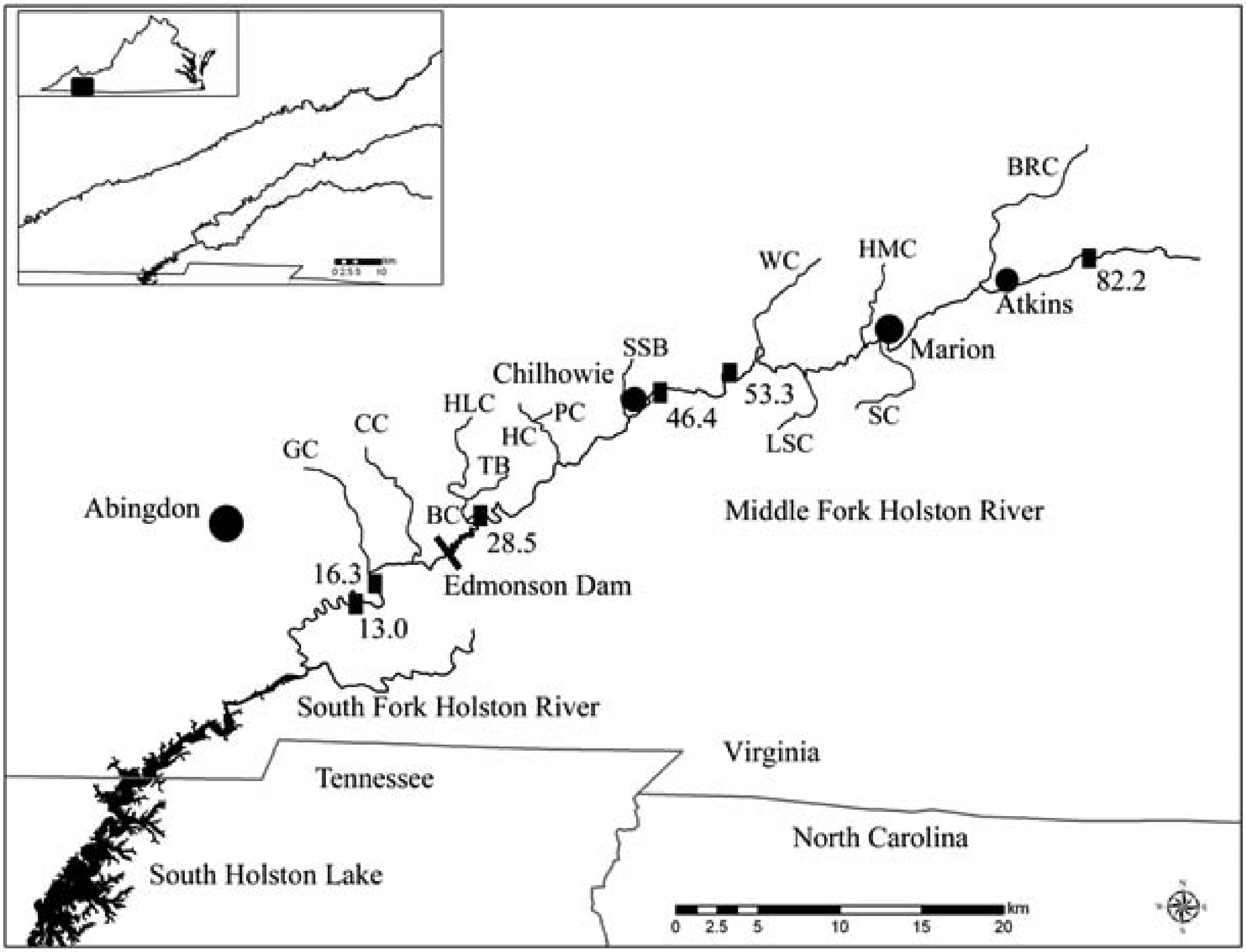

FIGURE 1

Mussel survey locations (indicated by squares and river kilometer) in the Middle Fork Holston River, Virginia, in 2010 and 2011. Circles indicate location of major towns. Tributaries are Greenway Creek (GW), Cedar Creek (CC), Hall Creek (HLC), Tattle Branch (TB), Byers Creek (BC), Hutton Creek (HC), Plum Creek (PC), Sulphur Springs Branch (SSB), Walker Creek (WC), Laurel Springs Branch (LSB), Hungry Mother Creek (HMC), Staley Creek (SC), and Bear Creek (BRC) (see impaired reach and tributary listings in Appendix 2.

We sampled freshwater mussels at six sites in the MFHR in 2010 and 2011 (Table 1; Fig. 1). Sample methods were similar to Henley et al. (1999). Mussel abundance at each site was measured in two ways: visual survey and quadrat sampling. These two methods were used because visual surveys cover more area and thus provide better estimates of site richness, but quadrat sampling provides better estimates of mussel density and size structure (Vaughn et al., 1997; Strayer & Smith, 2003). Survey crews consisted of at least two trained biologists for both methods.

TABLE 1

Study sites and results of mussel sampling in the Middle Fork Holston River, Virginia, during 2010. CPUE = catch per unit effort. Asterisks indicate that the absence of mussels in this study was due to bridge construction and prior mussel relocation (see text).

					Visual sampling		Quadrat sampling		
Site	County	Latitude (N)	Longitude (W)	Reach Length (m)	Total search time (person h)	Mussel CPUE(no./h)	No. of Transects	No. of Quadrats	Mussel density (no./m^2)
82.8	Smyth	36°53’19.08”	81°20’48.79”	60	1.67	0.00*	14	65	0.00*
53.3	Smyth	36°49’12.04”	81°37’08.08”	40	4.00	0.50	9	36	0.44
46.4	Smyth	36°50’14.36”	81°30’30.16”	50	3.40	2.34	15	60	0.07
28.5	Washington	36°50’06.38”	81°35’43.79”	120	17.10	6.20	24	96	0.33
16.3	Washington	36°44’23.98”	81°46’53.46”	40	2.00	88.00	9	36	1.67
13.0	Washington	36°41’32.28”	81°51’53.85”	45	2.00	0.00	10	40	0.10

Visual surveys were conducted using mask and snorkel to search the river bottom for mussels. During the searches, moderate-sized cobbles were overturned to locate mussels. When a live mussel was observed, its position in the substrate was marked by a flag. After visual survey at a site was complete, mussels marked by flags were removed, identified to species, measured for length, and returned to their original substrate position. The average duration of visual surveys across sites was 5.0 person-h (range = 1.7 – 17.1), and mussel abundance was expressed as catch per unit effort (CPUE, number/h).

Quadrat sampling was conducted with 0.25 m^2 quadrats positioned along transect lines perpendicular to the river channel. We sampled 36-96 quadrats on 9-24 transects at each site depending on site length (Table 1). The position of the first transect at each site was determined by selecting an arbitrary starting position at the downstream end of the sample reach, and then using a random number table to determine the number of paces upstream from the starting position for placement of the first transect. All subsequent transects were placed at 5 m intervals in an upstream direction. Four quadrats were randomly placed along each transect using a random number table, and quadrats were excavated to hardpan or to approximately 25 cm. Substrate from quadrats was not sieved, but we attempted to examine excavated substrate carefully for the presence of juvenile or other small mussels. Mussel densities were expressed as number/m^2. Mussels in each quadrat were identified to species, measured for length, and replaced at the point of collection.

We used a generalized linear mixed model (GLIMMIX, SAS Institute Incorporated, Cary, North Carolina) to test for statistical differences in mean density between 1998 amd 2010 at each site. The response variable (site mussel density) was designated as having a Poisson distribution with transects and quadrats set as random variables (quadrats within transects). The quadrat data provided adequate fit to the Poisson distribution (SAS generalized linear model, GENMOD: df=532, deviance X^2=561.12, p=0.185). We compared mean river-wide CPUE between 1998 and 2010 using natural log-transformed CPUE data and a paired t-test (Minitab 16, Minitab Incorporated, College Station, Pennsylvania). In previous surveys, including 1998, *Lasmigona holstonia* was only found in the headwaters of the river near MFHRKM 82.8 (Table 2). After conducting our survey in 2010, we learned that bridge reconstruction had occurred at this site in 2002, and an effort was made at that time to translocate as many individuals of this species as possible to another nearby site (Mair & Neves, 2002). This may explain the absence of *L. holstonia* in our survey at MFHRKM 82.8. Therefore, data from MFHRKM 82.8 were not included in the statistical analyses.

TABLE 2

Historical changes in mussel species richness in the Middle Fork Holston River.

Species	Study				
	Ortmann (1918)	Stansbery and Clench (1974)	Neves et al. (1980)	Henley et al. (1999)	This study
Actinonaias ligamentina			X		
Actinonaias pectorosa		X	X	X	X
Alasmidonta marginata	X	X	X		
Alasmidonta viridis	X				
Cyclonaias tuberculata		X		X	X
Elliptio dilatata	X	X	X	X	X
Epioblasma f. aureola	X	X	X	X	
Lampsilis fasciola	X	X	X	X	X
Lampsilis ovata		X	X		
Lasmigona costata	X	X	X	X	
Lasmigona holstonia		X		X	
Medionidus conradicus	X	X	X	X	
Pegias fabula		X			
Pleurobema oviforme	X	X	X	X	X
Pleuronaia barnesiana	X	X	X	X	X
Pleuronaia dolabelloides		X	X	X	X
Ptychobranchus fasciolaris		X	X	X	X
Ptychobranchus subtentum	X	X	X	X	X
Villosa iris	X	X	X	X	X
Villosa vanuxemensis	X	X	X	X	X
Total Species	12	18	16	15	11

Two types of data from the Virginia Department of Environmental Quality (VDEQ) were obtained to assess potential causes of mussel declines in the MFHR: a list of reaches in the MFHR and tributaries impaired for recreational and/or aquatic life uses under the criteria of sections 303(d) and 305(b) of the US Clean Water Act (VDEQ, 2010; 2012), and results of sediment contaminant analyses from two sites (T. Frasier, VDEQ, Abingdon, Virginia, unpublished data). Sites with sediment analysis were MFHRKM 16.3 (only metals analyzed from one sediment sample from 2008) and 42.0 (metals and organics contaminants analyzed from 20 collection dates from 1981 to 1998). Although MFHRKM 42.0 was not one of our survey sites, sediment results from this site probably represent past activities from upstream locations in the watershed, including the towns of Atkins, Marion, and Chilhowie, Virginia. Sediment samples were analyzed for metals and organic compounds by the Division of Consolidated Laboratory Services, Department of General Services, Richmond, Virginia, using USEPA methods for sample preparation (3005A) and analysis (200.8) (USEPA, 1992; 1994). We reported only contaminants that were above detection limits from the VDEQ sediment database. We compared sediment contaminant concentrations measured at MFHRKM 16.3 and 42.0 with the consensus-based freshwater sediment quality guidelines of MacDonald et al. (2000), and when guidelines were not provided for a certain contaminant, freshwater sediment-screening benchmarks of USEPA (2006) were used.

RESULTS

Mussel species richness in the MFHR has declined substantially in recent decades (Table 2). Ortmann (1918) surveyed only 2 sites in the MFHR in 1912 and 1913 but found 12 species. Stansbery and Clench (1974) surveyed 21 sites in the late 1960s and early 70s and found 18 species, and Neves et al. (1980) collected 17 species at nine sites in the late 1970s; together, these surveys reported 20 species present in the river prior to 1980. In 1998, Henley et al. (1999) found 15 species, but we found only 11 species in 2010, representing roughly half of historical richness. Only one site yielded no mussels (MFHR 82.8; see Methods). At each of our five study sites where mussels were observed, richness was approximately half that observed in 1998 (Fig. 2; Appendix 1), and several species were extremely rare.

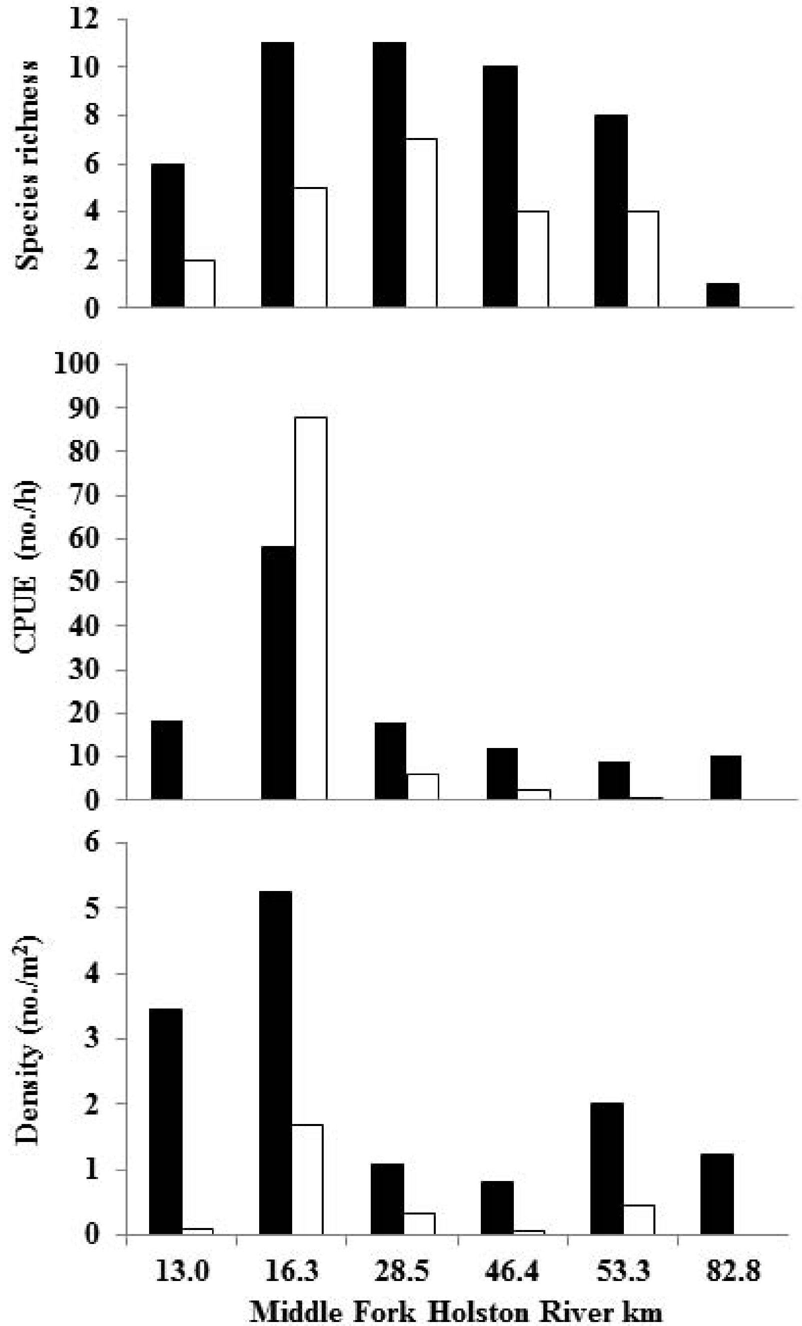

FIGURE 2

Changes in mussel assemblages at six sites in the Middle Fork Holston River (MFHR) from 1998 to 2010. Black bars are results of 1998 surveys and white bars are 2010. No mussels were found at MFHR kilometer location 82.8 in 2010 due to mussel relocation prior to our survey (see text).

Lampsilis fasciola and *Pleurobema oviforme* each were represented by single individuals, and *Elliptio dilatata* and *Pleuronaia barnesiana* were represented by only 4 and 5 individuals, respectively. Two species proposed for federal listing as endangered were collected during this survey: *Ptychobranchus subtentum* at MFHRKM 46.4 and *Pleuronaia dolabelloides* at MFHRKM 16.3 and 13.0. *Epioblasma f. aureola* was collected only at MFHRKM 28.5 in 1998, but the species was not observed in 2010. In subsequent sampling in 2013, one live *P. subtentum* was collected at MFHRKM 16.3 (one of our sample sites), and numerous *P. dolabelloides* were collected at MFHRKM 15.1 (a previously unsurveyed site; D. Schilling, personal communication).

Mussel abundance declined dramatically from 1998 to 2010 (Fig. 2; Appendix 1). Mussel abundance, as measured by CPUE, declined by ≥50% at all sites except for MFHR 16.3, where CPUE was higher in 2010 (88.0 mussels/h in 2010 versus 58.3/h in 1998). Mean (±SD) river-wide CPUE declined only slightly but significantly from 23.1±20.1/h in1998 to 19.4±38.4/h in 2010 (*ln*CPUE, n=5, t=2.45, p=0.035), but the small magnitude of this difference is a result of the high CPUE at MFHR 16.3 in 2010, which was nearly 20X higher than all other sites in 2010. River-wide mussel density declined more dramatically from 2.5±1.8/m^2 in 1998 to 0.5±0.7/m^2 in 2010 (df=1, F=36.55, p<0.0001). A few species appeared to show slight increases in abundance at some sites (Appendix 1), but the precision of these estimates is low, and these apparent increases probably are not statistically significant.

There was no evidence of recent recruitment at any of our survey sites. The mean size of each species indicated that all individuals were adults, and many individuals appeared to be of advanced age (Table 3). The smallest individual observed was a 28.5 mm *Villosa iris*. Subsequent sampling at MFHRKM 16.3 in 2013 found a few *Pleuronaia dolabelloides* that were estimated to be 7-10 years old (D. Schilling, personal communication).

TABLE 3

Mean and minimum lengths (in parentheses, both mm) of mussels collected in the Middle Fork Holston River in 2010. Values are from combined results of CPUE and quadrat sampling; integers below size measurements are the number of collected and measured individuals; if only one mussel was measured at a site, then no minimum size is provided. No mussels were found at MFHRKM 82.8 due to mussel relocation prior to our survey (see text).

	Middle Fork Holston River Sites (km)				
Species	**53.3**	**46.4**	**28.5**	**16.3**	**13.0**
Actinonaias pectorosa	-	-	110.3 (107.4) 3	118.8 (95.0) 124	-
Cyclonaias tuberculata	-	-	-	83.7 (75.0) 9	-
Elliptio dilatata	72.0 1	88.5 1	88.1 (86.1) 2	-	-
Lampsilis fasciola	46.1 1	-	-	-	-
Pleuronaia barnesiana	-	64.4 1	57.0 (52.3) 4	-	-
Pleuronaia dolabelloides	-	-	-	78.3 (57.0) 44	86.0 1
Pleurobema oviforme	-	-	57.2 1	-	-
Ptychobranchus fasciolaris	-	-	87.3 (70.6) 40	108.7 (98.0) 14	-
Ptychobranchus subtentum	-	100.4 (89.7) 6	-	-	-
Villosa iris	45.1 1	-	49.7 (28.5) 31	-	-
Villosa vanuxemensis	38.4 (32.4) 3	49.2 1	46.2 (34.0) 33	-	-

Twenty-six reaches of the MFHR and tributaries are classified as impaired for recreational and/or aquatic life uses by the VDEQ (VDEQ, 2010; 2012; Appendix 2). All of the sites we surveyed are in impaired reaches except MFHRKM 13.0. Causes of impairment in the watershed listed by VDEQ include sediment, *Escherichia coli*, fecal coliform, and alterations to benthic macroinvertebrate communities, all generally attributed to unrestricted livestock access to water bodies, animal feeding operations, crop production, grazing in riparian zones, and the category "rural (residential areas)" that presumably describes inadequate and failing residential septic systems (VDEQ 2010, 2012). Of particular importance to remaining mussel populations, including the federal candidate species *Pleuronaia dolabelloides* and *Ptychobranchus subtentum*, is the impaired reach that contains site MFHRKM 16.3 (VAS-O05R-MFHR3A00, recreational impairment), and two impaired tributaries that enter MFHR in this area (Greenway Creek, VAS-O05R-GRW01A02, aquatic life and recreational impairments; and Cedar Creek, VAS-O05-CED01A94 and VAS-O05-ECE01A02, aquatic life and recreational impairments) (Appendix 2). The reach of the MFHR that contains MFHRKM 46.4 where *P. subtentum* also was found is impaired (VAS-O04R-MFH01A00, recreational impairment). In impairment notes related to this reach, VDEQ states that DDT was detected in sediment samples.

Many contaminant concentrations measured in sediment at MFHRKM 42.0 were above suggested screening levels, but no metal concentrations were above these levels at MFHRKM 16.3 (Appendix 3). At MFHRKM 42.0, mean concentrations of antimony, iron, lead, manganese, and selenium were above sediment quality guidelines (SQG); however, concentrations of arsenic, cadmium, mercury, nickel, zinc, total polychlorinated biphenyls (PCBs), and dichlorodiphenyldichloroethylene (DDE, breakdown byproduct of DDT) also periodically rose above SQGs (Appendix 3). The contaminant levels measured from sediment from MFHRKM 42.0 result from upstream activities, including those in and around the towns of Atkins, Marion, and Chilhowie.

DISCUSSION

The mussel fauna of the MFHR has declined dramatically since 1998 in both species richness and abundance, and the lack of recruitment portends a further diminishment of the fauna. However, this decline appears to have begun prior to 1998 judging by the meager evidence of recruitment and the disappearance of several species at the time of the Henley et al. (1999) survey. The decline of the MFHR fauna is yet another example of an unexplained, enigmatic mussel decline characterized by a gradual erosion of mussel diversity apparently due to curtailment of recruitment (Haag, 2012). It is noteworthy that most species that have disappeared from the MFHR are short-lived (life span < about 20 y; e.g., *Alasmidonta* spp., *Epioblasma florentina aureola*, *Lasmigona* spp., *Medionidus conradicus*, *Pegias fabula*) (Haag & Rypel, 2010). The remaining fauna is composed mainly of long-lived species (lifespan > 30 y) such as *Actinonaias pectorosa*, *Cyclonaias tuberculata*, *Elliptio dilatata*, *Pleuronaia* spp., and *Ptychobranchus fasciolaris*, but the uniformly large size of these individuals suggests that they recruited prior to the appearance of factors that now limit recruitment. As in other streams that have experienced enigmatic mussel declines, factors responsible for the lack of recruitment in the MFHR are unknown. Sediment and fecal bacteria inputs are sources of use impairment in the MFHR. The river is prone to extended periods of high turbidity after rain events, and in our experience, the water clears much more slowly after these events than in other streams in southwestern Virginia. No studies have determined primary sources of these contaminants, but our observations suggest that unrestricted cattle access and erosion in riparian zones are major causes. This also is concordant with the fact that 41% of the watershed is pastureland (USEPA, 2010). The effect of fecal bacteria on mussel survival is unknown, and there is little evidence for a direct negative effect of sediment (Haag, 2012). However, extended periods of suspended solids, as seen in the MFHR, can cause sharply reduced fertilization of mussel eggs (Gascho Landis et al., 2013).

Both of these factors may be indirectly involved in mussel declines via their role in increasing ammonia concentrations in stream sediments. Juvenile mussels are highly sensitive to ammonia (Augspurger et al., 2003; Geist & Auerswald, 2007; Wang et al., 2007a, 2007b). Animal manure is a major source of nutrient enrichment and eutrophication in streams, which also can lead to elevated ammonia, and unrestricted cattle access to streams is linked to increased total nitrogen, total phosphorus, total suspended solids, ammonium, turbidity, and *E. coli* (Vidon et al., 2008). Sedimentation can further exacerbate ammonia levels in the streambed by reducing interstitial sediment oxygen concentrations, which in turn reduces the ability of nitrifying bacteria to convert ammonia to less toxic nitrates. Because juvenile mussels reside primarily in and feed on sediments, they may be inordinately exposed to elevated ammonia in the streambed (Cope et al., 2008; Strayer & Malcom, 2012). Testing of sediment ammonia and oxygen concentrations in the MFHR is urgently needed to evaluate this potential cause of mussel declines. In addition, a wide variety of landowner incentive programs designed to restrict cattle access to streams are available through agencies such as the National Resource Conservation

Service, and promotion of these programs could help reduce this source of contamination in the watershed.

The available sediment data are not contemporary with our study period, but the persistence of metals, PCBs, DDE, and other compounds in sediment suggests that they still may limit juvenile survival. Resuspension and reoxygenation of these sediment contaminants during sustained turbidity events also may contribute to their continued bioavailability (Eggleton & Thomas, 2004). In addition to juvenile mortality, chronic exposure may result in sublethal effects to remaining adult mussels in the MFHR. Many of these compounds can negatively affect gamete production and quality and larval survival in other organisms including marine bivalves (Bayne et al., 1981; McDowell et al., 1999; Pocar et al., 2003; Tay et al., 2003; Lewis & Ford, 2012), but their effects on freshwater mussels are largely unknown.

Sediment data show considerable variation in contaminant levels among sites. Metals were not above screening guidelines at MFHRKM 16.3, but many exceeded screening levels at MFHRKM 42.0. Edmonson Dam (MFHRKM 22.5) and DeBusk milldam (MFHRKM 31.6) may act as settling basins that intercept many contaminants before they reach the lower river. This phenomenon may partially explain the higher mussel abundance that we observed at MFHRKM 16.3 in the lower river, but the lack of recruitment even at this site indicates the presence of significant stressors throughout the river.

The MFHR is an important refuge for the diverse Tennessee River basin mussel fauna, and identification and remediation of specific factors responsible for mussel declines are urgently needed. Mitigation efforts throughout the river should to be guided by results of sediment and pore-water contaminant analyses. Therefore, we recommend that Virginia state agencies coordinate the collection of sediment and interstitial water samples at sites along the length of the river for determinations of organic and inorganic contaminants. Possible links among these data and current industrial discharges in the watershed need to be determined. We also suggest immediate implementation of best management practices (BMPs, e.g., riparian restoration, fencing, and alternative water sources) that would reduce nutrient and sediment inputs. An overarching motivation for these efforts is that the drastic decline in mussel species richness and abundance is an indicator of highly degraded conditions in the river and its watershed, which affects all stakeholders in the region. Without determination of specific stressors and appropriate mitigation, the mussel fauna of the MFHR likely will disappear completely as remaining individuals become senescent and die.

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APPENDIX 1

Species relative abundance and density at six sites in the MFHR in 1998 (Henley et al., 1999) and 2010 (this study). Top values for each species are CPUE (number/h) and bottom values are density (number/m^2). Asterisks indicate that the absence of mussels in this study was due to bridge construction and prior mussel relocation (see text). Dashes indicate species not observed.

Species	Middle Fork Holston River Sites (km)											
	82.8		53.3		46.4		28.5		16.3		13.0	
	1998	2010	1998	2010	1998	2010	1998	2010	1998	2010	1998	2010
Actinonaias pectorosa	-	-	-	-	-	-	-	0.18 0.00	48.4 2.88	59.00 0.67	3.53 0.20	-
Cyclonaias tuberculata	-	-	-	-	-	-	-	-	0.49 0.00	3.50 0.22	0.94 0.12	-
Elliptio dilatata	-	-	2.86 0.88	0.25 0.00	1.00 0.08	0.29 0.00	0.17 0.00	0.12 0.00	-	-	-	-
Epioblasma f. aureola	-	-	-	-	-	-	0.17 0.00	-	-	-	-	-
Lampsilis fasciola	-	-	0.14 0.00	0.00 0.11	0.50 0.00	-	1.17 0.00	-	0.25 0.00	-	-	-
Lasmigona costata	-	-	-	-	-	-	-	-	0.50 0.00	-	-	-
Lasmigona holstonia	10.3 1.24	* *	-	-	-	-	-	-	-	-	-	-
Medionidus conradicus	-	-	-	-	1.00 0.00	-	2.00 0.08	-	-	-	-	-
Pleurobema oviforme	-	-	3.29 0.44	-	6.00 0.28	-	8.50 0.40	0.06 0.00	1.47 0.24	-	2.11 0.00	-
Pleuronaia barnesiana	-	-	0.14 0.12	-	0.50 0.12	0.29 0.00	1.50 0.12	0.23 0.00	0.98 0.12	-	-	-
Pleuronaia dolabelloides	-	-	0.29 0.00	-	0.50 0.00	-	1.00 0.00	-	5.66 1.76	19.50 0.56	11.5 3.12	0.00 0.10
Ptychobranchus fasciolaris	-	-	-	-	-	-	0.83 0.12	2.16 0.13	0.00 0.24	6.00 0.22	0.24 0.00	-
Ptychobranchus subtentum	-	-	-	-	0.50 0.12	1.47 0.07	-	-	0.25 0.00	-	-	-
Villosa iris	-	-	0.14 0.00	0.00 0.11	0.50 0.08	-	1.17 0.20	1.58 0.17	0.25 0.00	-	-	-
Villosa vanuxemensis	-	-	2.14 0.56	0.25 0.22	1.50 0.12	0.29 0.00	1.00 0.16	1.87 0.04	-	-	-	-
Site Total	10.3 1.24	* *	9.00 2.00	0.50 0.44	12.00 0.80	2.34 0.07	18.01 1.08	6.20 0.33	58.25 5.24	88.00 1.67	18.32 3.44	0.00 0.10

APPENDIX 2

Stream reaches in the Middle Fork Holston River (MFHR) watershed listed as impaired by Virginia Department of Environmental Quality under sections 303 (d) and 305 (b) of the Clean Water Act (VDEQ, 2010; 2012). In the VDEQ database, all assessment unit labels are preceded by the prefix 'VAS-'; note that Cedar and Staley creeks each have two impaired assessment units. In columns under "Impairment Cause", the first value is the length (km) of the impaired reach, and the second value (in parentheses) is the actual or target date for Total Maximum Daily Load (TMDL) development. Cause category 4A indicates that TMDL is not necessary or previously developed, and 5A indicates that TMDL is required. Impairment causes are determined by exceedance of impairment thresholds. Benthic impairment cause was determined using the Virginia Stream Condition Index (VDEQ, 2010). Tributary locations are shown on Figure 1.

				Impairment Cause				Use Impairment	
Assessment Unit (MFHR reach)	Stream	County	Cause Category	*Escherichia coli*	Fecal coliform	Benthic	Sediment	Recreation	Aquatic Life
O05R_BYS01A94	Byers Creek	Washington	4A	0.8 (2002)	-	0.8 (2004)	-	√	√
O05R_CED01A94 O05R_ECE01A02	Cedar Creek	Washington	4A	10.5 (2002)	3.0 (2002)	13.5 (2004)	13.5 (2004)	√	√
O05R_HAL01A94	Hall Creek	Washington	4A	-	10.9 (2002)	10.9 (2004)	10.9 (2004)	√	√
O05R_HTO01A94	Hutton Creek	Washington	4A	7.7 (2002)	-	7.7 (2004)	7.7 (2004)	√	√
O05R_MFHR04A00	MFHR	Smyth & Washington	4A	-	14.7 (2014)	14.7 (2020)	-	√	√
O05R_MFHR03A00	MFHR	Washington	4A	4.2 (2014)	-	-	-	√	-
O05R_MFHR05A00	MFHR	Washington	4A	-	5.9 (2014)	5.9 (2014)	-	√	√
O03R_MFHR01A00	MFHR	Smyth, Washington, & Wythe	4A	28.9 (2014)	8.8 (2014)	-	-	√	-
O03R_MFHR02A00	MFHR	Smyth, Washington, & Wythe	4A	8.3 (2014)	8.3 (2014)	-	-	√	-
O03R_MFHR05A04	MFHR	Smyth, Washington, & Wythe	4A	5.2 (2022)	5.9 (2014)	5.9 (2014)	-	√	√
O03R_MFHR04A98	MFHR	Smyth, Washington, & Wythe	4A	-	6.8 (2014)	-	-	√	-
O05R_PLU01A02	Plum Creek	Washington	4A	-	3.5 (2002)	3.5 (2004)	-	√	√
O05R_TAT01A02	Tattle Branch	Washington	4A	-	4.4 (2002)	4.4 (2004)	4.4 (2004)	√	√
O05R_XCG01A02	Unnamed tributary to Hall Creek	Washington	4A	-	2.7 (2002)	2.7 (2004)	2.7 (2004)	√	√
O05R_XCD01A02	Unnamed tributary to Hutton Creek	Washington	4A	-	6.5 (2002)	6.5 (2004)	6.5 (2004)	√	√
O05R_XDY01A08	Unnamed tributary to MFHR	Washington	4A	1.4 (2020)	-	-	-	√	-
O05R_CWF01A02	West Fork Cedar Creek	Washington	4A	-	2.5 (2002)	2.5 (2004)	2.5 (2004)	√	√
O03R_BER01A02	Bear Creek	Smyth	5A	9.0 (2022)	-	-	-	√	-
O03R_BER02A04	Bear Creek	Smyth	5A	-	-	6.5 (2020)	-	-	√
O05R_GRW01A02	Greenway Creek	Washington	5A	7.8 (2020)	-	7.8 (2022)	-	√	√
O04R_HUN02A02	Hungry Mother Creek	Smyth	5A	7.8 (2018)	-	-	-	√	-
O04R_LRL1A04	Laurel Springs Creek	Smyth	5A	3.3 (2018)	-	-	-	√	-
O03R_STA01A02 O03R_STA01B10	Staley Creek	Smyth	5A	11.0 (2022)	-	-	-	√	-
O04R_SUL01A12	Sulphur Springs Branch	Smyth	5A	11.0 (2024)	-	-	-	√	-
O04R_WAL01A02	Walker Creek	Smyth	5A	20.6 (2018)	-	-	-	√	-

APPENDIX 3

Mean (±SE) contaminant concentrations in sediment collected by Virginia Department of Environmental Quality (VDEQ) at Middle Fork Holston River kilometer locations 42.0 (Washington-Smyth county line approximately 2.4 km downstream of Chilhowie, Virginia; 20 collection dates during 1981 to 1998) and 16.3 (see Table 1 for location; one collection date in 2008). *n* is the number of observations that were above detection limits. Sediment screening levels are criteria used to evaluate the risk of an observed contaminant concentration to aquatic organisms as follows. TEC is the consensus-based threshold effects concentration below which harmful effects on sediment-dwelling organisms are not expected (MacDonald et al., 2000; clarification of TEC for selenium provided by A. D. Lemly, USDA Forest Service and Department of Biology, Wake Forest University, Winston-Salem, North Carolina, personal communication). PEC is the consensus-based probable effects concentration above which harmful effects on sediment-dwelling organisms are expected to occur frequently (MacDonald et al., 2000). FSSB are freshwater sediment screening benchmarks used to evaluate sediment data from Superfund sites and to classify ecological risk (USEPA, 2006). Effect range – low (ERL) is the concentration below which adverse effects would be rarely observed (MacDonald et al., 2000). Lowest effect level (LEL) is the concentration below which no effects on the majority of sediment-dwelling organisms are expected (MacDonald et al., 2000). FSSB criteria were used only when TEC or PEC were not available.

	Site							Sediment Screening Levels		
	42.0				16.3					
Analyte	Mean	SE	Range	*n*	Mean	SE	*n*	TEC	PEC	FSSB
Metals (mg/kg DW)										
Aluminum	13,584	1,270	9,390 – 17,400	5	8,620	-	1	-	-	-
Antimony	20.0	13.0	7.0 – 33.0	2	-	-	-	-	-	2.0[ERL]
Arsenic	7.3	0.8	2.0 -13.2	11	-	-	-	9.79	33.0	-
Cadmium	0.8	0.3	0.2 – 2.5	7	-	-	-	0.99	4.98	-
Chromium	21.8	1.3	13.9 – 31.0	14	17.8	-	1	43.4	111	-
Copper	21.8	1.6	10.1 – 31.0	14	15.0	-	1	31.6	149	-
Iron	21,170	978	18,700 – 23,700	5	15,200	-	1	-	-	20,000[LEL]
Lead	36.0	5.2	13.0 – 87.5	14	22.4	-	1	35.8	128	-
Manganese	650.2	63.1	495 – 819	5	358.0	-	1	-	-	460[LEL]
Mercury	0.10	0.03	0.10 – 0.30	6	-	-	-	0.18	1.06	-
Nickel	20.0	1.1	9.6 – 25.0	14	13.7	-	1	22.7	48.6	-
Selenium	4.0	2.0	2.1 – 6.0	2	-	-	-	-	-	2.0[TEC]
Thallium	1.6	0.6	1.0 – 2.1	2	-	-	-	-	-	-
Zinc	102.9	6.9	76.7 – 146.0	13	89.2	-	1	121	459	-
Total PCBs (μg/kg DW)	25.8	24.8	1.0 – 100.0	4	-	-	-	59.8	676	-
DDE (μg/kg DW)	6.8	6.7	0.1 – 40.0	6	-	-	-	3.16	31.3	-

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MOLLUSK CONSERVATION SOCIETY

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WALKERANA The Journal of the Freshwater Mollusk Conservation Society

OUR PURPOSE

The Freshwater Mollusk Conservation Society (FMCS) is dedicated to the conservation of and advocacy of freshwater mollusks, North America's most imperiled animals. Membership in the society is open to anyone interested in freshwater mollusks who supports the stated purposes of the Society which are as follows:

1) Advocate conservation of freshwater molluscan resources;

2) Serve as a conduit for information about freshwater mollusks;

3) Promote science-based management of freshwater mollusks;

4) Promote and facilitate education and awareness about freshwater mollusks and their function in freshwater ecosystems;

5) Assist with the facilitation of the National Strategy for the Conservation of Native Freshwater Mussels (Journal of Shellfish Research, 1999, Volume 17, Number 5), and a similar strategy under development for freshwater gastropods.

OUR HISTORY

The FMCS traces it's origins to 1992 when a symposium sponsored by the Upper Mississippi River Conservation Committee, USFWS, Mussel Mitigation Trust, and Tennessee Shell Company brought concerned people to St. Louis, Missouri to discuss the status, conservation, and management of freshwater mussels. This meeting resulted in the formation of a working group to develop the National Strategy for the Conservation of Native Freshwater Mussels and set the ground work for another freshwater mussel symposium. In 1995, the next symposium was also held in St. Louis, and both the 1992 and 1995 symposia had published proceedings. Then in March 1996, the Mississippi Interstate Cooperative Research Association (MICRA) formed a mussel committee. It was this committee (National Native Mussel Conservation Committee) whose function it was to implement the National Strategy for the Conservation of Native Freshwater Mussels by organizing a group of state, federal, and academic biologists, along with individuals from the commercial mussel industry. In March 1998, the NNMCC and attendees of the Conservation, Captive Care and Propagation of Freshwater Mussels Symposium held in Columbus, OH, voted to form the Freshwater Mollusk Conservation Society. In November 1998, the executive board drafted a society constitution and voted to incorporate the FMCS as a not-for-profit society. In March 1999, the FMCS held it's first symposium "Musseling in on Biodiversity" in Chattanooga, Tennessee. The symposium attracted 280 attendees; proceedings from that meeting are available for purchase. The second symposium was held in March 2001 in Pittsburgh, Pennsylvania, the third in March 2003 in Raleigh, North Carolina, the fourth in St. Paul, Minnesota in May 2005, the fifth in Little Rock, Arkansas in March 2007, the sixth in Baltimore, Maryland in April 2009, the seventh in Louisville, Kentucky in 2011, and the eighth in Guntersville, Alabama in 2013. The society also holds workshops on alternating years, and produces a newsletter four times a year.

FMCS SOCIETY COMMITTEES

Participation in any of the standing committees is open to any FMCS member. Committees include:

- Awards
- Environmental Quality and Affairs
- Gastropod Distribution and Status
- Genetics
- Guidelines and Techniques
- Information Exchange - Walkerana and Ellipsaria
- Mussel Distribution and Status
- Outreach
- Propagation and Restoration

TO JOIN FMCS OR SUBMIT A PAPER

Please visit our website for more information at http://www.molluskconservation.org

Or contact any of our board members or editors of WALKERANA to talk to someone of your needs. You'll find contact information on the back cover of this publication.